Journal of Forestry Research (2013) 24(2): 255–262
DOI 10.1007/s11676-012-0316-7

ORIGINAL PAPER

# Forest structure and woody plant species composition after a wildfire in beech forests in the north of Iran

Mohammad Naghi Adel • Hassan Pourbabaei • Ali Omidi • Daniel C Dey

Received: 2011-12-04;  Accepted: 2012-03-20


**Abstract**: Beech (*Fagus orientalis* Lipsky) forest covers about 565,000 ha of land in Guilan province, north of Iran and forms a major carbon pool. It is an important economic, soil protection and recreation resource. We studied long-term effects of fire on the structure and composition 37 years after fire occurrence in these forests. To do this research, we selected 85 ha burned and 85 ha unburned beech forests). The results indicated that the fire had not changed the overall uneven-aged structure, but it changed forest composition from pure stands to mixed stands that now include species such as *Carpinus betulus*, *Acer cappadocicum* and *Alnus subcordata*. The density of trees and regeneration was significantly increased, while the density of shrubs significantly decreased. The main reasons for increased tree regeneration were attributed to (1) reduction of litter depth, and (2) increase in available light from opening of the canopy and reduction in shrub competition. It is apparent that the forest is on a path to return to its natural state before the fire after 37 years.

**Keywords:** fire, structure, composition, regeneration, beech forest, Guilan Province

## Introduction

The total area of forest in Iran is 12 million ha, accounting for 8% of the total land area of Iran. Approximately 1.8 million ha of these forests are located in northern Iran (i.e., the Hyrcanian Forests) on the northern slopes of the Alborz Mountains overlooking the Caspian Sea (Sagheb-Talebi et al. 2004). About 1.2 million ha of these forests are commercial and the other parts are protected or conservation forests. Forests of northern Iran that are located in Hyrcanian regions are known as Hyrcanian forests. Hyrcanian forests are of the broadleaf deciduous type, with local inclusions of Mediterranean forest types. Distributions of forests in Hyrcanian are not homogeneous, and forest productivity (in terms of growth rates) decreases from west to east. Forests of the Guilan Province are located in the western part of the Hyrcanian forest region. Caspian forests (Hyrcanian forests) have apparently a high similarity to broadleaf forests of central Europe, north of Turkey and Caucasus (Mohadjer 2006). The dominant species in the northern forests of Iran is beech (*Fagus orientalis* Lipsky). Beech forests are the richest forest community in Iran because they are both economically and environmentally valuable. The greatest forest volume occurs in Iran's beech forests (Taheri and Pilehvar 2008).

Forests dominated by beech (*F. orientalis*) species cover about 565 thousand hectares of land and account for the total area of indigenous forest in Guilan. The beech forest thus forms a major carbon pool (Hall et al. 2001) and is an important economic, soil protection and recreation resource (Wardle 1984). Guilan beech species are poorly adapted to fire and even low temperature burns can lead to forest destruction. Beech biology, ecology and forest history lead to the hypothesis that the elimination of the beech canopy shading by fire, concurrent with the burning of the allelopathic litter, may create an opportunity for tree species diversity (van Gils et al. 2009).

Disturbances can cause major changes in plant communities depending on their intensity, extent, frequency, seasonality, and the resilience of the component species (Levin and Paine 1974; Connel 1978; Huston 1994; Grime 2001; Ross et al. 2004; Coates et al. 2006; Herath et al. 2009). Fire is one of the most important disturbance factors in natural ecosystems throughout the world (Moretti and Barbalat 2004) and prescribed fire plays a key role in natural resource management (Keeley et al. 2003; Huang et al. 2007; Yang et al. 2008). Fire disturbances often help

The online version is available at http://www.springerlink.com

Mohammad Naghi Adel (✉) • Hassan Pourbabaei
Facultad de Department of Forestry, Natural Resources Faculty , University of Guilan, Somehsara, P. O. Box 1144, Guilan, Iran.
Tel: +98-1823220895; Email addresses: mn.adel87@yahoo.com.

Ali Omidi
Department of Forestry, Natural Resources Office, Guilan, Iran.

Daniel C Dey
USDA Forest Service, Northern Research Station, 202 Natural Resources Building, Columbia, MO, 65211, USA.

Responsible editor: Chai Ruihai

to maintain plant species diversity (Gill 1981; Bell et al. 1984. Fox and Fox 1986b; Lamont et al. 2001; Blackhall, et al. 2008), and in fact, fire disturbance is a major factor driving patterns of vegetation structure and composition in natural ecosystem (Whelan 1995; Gonzales-Tagle et al. 2008). The degree of change from pre-fire community to post-fire community is influenced by the intensity, severity, periodicity, and seasonality of the fire (Wright and Bailey 1982), as well as other factors like precipitation cycles (Anderson et al. 1968; Moore et al. 2006) and grazing patterns (Krueger and Winward 1974; Madany and West 1983; Zimmerman and Neuenschwander 1984). Fires in northern Iran normally occur in autumn when hot-dry winds and short periods of drought cause forest floor litter to dry. These are mostly surface fires that rarely exceed 10–30 cm in flame height under normal fuel and humidity conditions. Fire burns 300–400 ha of forest annually in these forests (Banj-Shafiei et al. 2010).

Many forest studies have examined short-term changes in understoty vegetation following a wildfire; however, very few long-term studies are available (Bataineh et al. 2006). Also, analyses of wildfire effects are rare because it is difficult to study random events with robust, replicated experiments (van Mantgem and Schwarts 2001). Therefore, most studies of fire effects in forests have focused on prescribed burns that are more easily controlled and manipulated (e.g. Harris and Covington 1983; Armour et al. 1984; Andariese and Covington 1986; Reinhardt and Ryan 1988; Moreira et al. 2003; Fernandez-Abascal et al. 2004). Knowledge about the effects of wildfire would greatly increase our understanding of the role of fire in forests and would help in management decisions concerning the appropriate application of fire (Laughlin et al. 2004).

Despite the potential effects of fires burning in the most productive forest region in Iran, there has not been any study on fire impacts on forest structure, composition, and regeneration, which is fundamental to forest sustainability. Research on fire-caused changes on Guilan forests can provide information about forest vegetation after fire. Based on the information, forest managers can project future forest conditions, evaluate the desirability of allowing fires to burn, determine forest degradation due to fire, and formulate fire management plans. The aim of our study was to investigate the long-term impacts of natural fires on structure, composition and regeneration. Specific objectives focused on (1) fire effect on forest structure, (2) fire effect on density and species composition in the tree and shrub layer, (3) fire effect on regeneration?

## Materials and methods

### Study area

The study area is located in the Zilaki, at Roudbar city in the south of Guilan province in northern Iran (36°54′30″ to 36°56′6″ N latitude and 49°46′24″ to 49°51′17″ longitude E) (Fig.1). Elevation within the study area ranges from 1010 to 1560 m a.s.l., (by means of Thommen altimeter), slopes are between 30% and 40%, (by means of Suunto clinometer), and the general aspect is north (by means of Suunto compass).. Common forest soils are deep brown, heavy texture, with weak acidic pH (6.2–6.8). Type of parent material is lime silt, sandstone, siltstone and shill. The maximum temperature is 29.27°C in August and minimum temperature is 2.74°C in February. In Hyrcanian forests, harvesting is most commonly done by the single-tree selection method. Although the area is under a forest management plan, there has not been any harvesting in the study area due to a lack of a forest road network. The forest is uneven-aged with a mixture of deciduous broadleaf species but *Fagus orientalis* Lipsky is dominant in the forests. A fire with high severity occurred in December 1973, burning 85 ha of forest in seven days.

### Data collection

This study included both burned (B) and unburned (UB) areas of 85 hectares each. The UB area was located far from (ca. 100 m) the B area to avoid direct or indirect effects from the fire. The two areas had similar elevations, slopes and aspects. Within each of the study areas we used a random systematic 150 m × 200 m sampling grid to locate 30 circular plots (each plot is 1,000 $m^2$) resulting in a total of 60 plots (Fig.1). In each plot, we recorded environmental factors such as topographic aspect, slope percentage, elevation, crown canopy percentage, herbal layer cover percentage, and species. Diameter at breast height (DBH) of those trees more than 7.5 cm in diameter was measured, and number of shrubs were counted. For the regeneration assessment, we used four subplots within each larger plot. Each subplot was 25 $m^2$ and was located along one of the four cardinal directions within the larger plot. All seedlings and saplings were tallied as follows: Class 1 = height < 1.3 m, Class 2 = DBH < 2.5 cm, and Class 3 = 2.5 cm < DBH < 7.5 cm. In each plot, litter depth at 5 points was measured. Data collection was done in July 2010, 37 years after the fire in 10 days.

### Data analysis

The Kolmogorov–Smirnov test was used to evaluate the normality of parameters. Non-paired t tests were used to compare data that were normally distributed. For parameters that were not normally distributed, the nonparametric Mann-Whitney U test was used.

## Results

### Tree layer

There was a significant difference ($p \leq 0.05$) between the B and UB areas with regard to percentage of canopy cover, herbaceous percentage, number of trees per hectare, litter depth, mean DBH and basal area (Table 1). Percentage of canopy cover, litter depth, mean of DBH and basal area were greater in the UB area. On the other hand, percentage of herbaceous cover and number of trees per hectare were greater in the B area. There was not any significant difference between these two areas regarding the number of

dead trees.

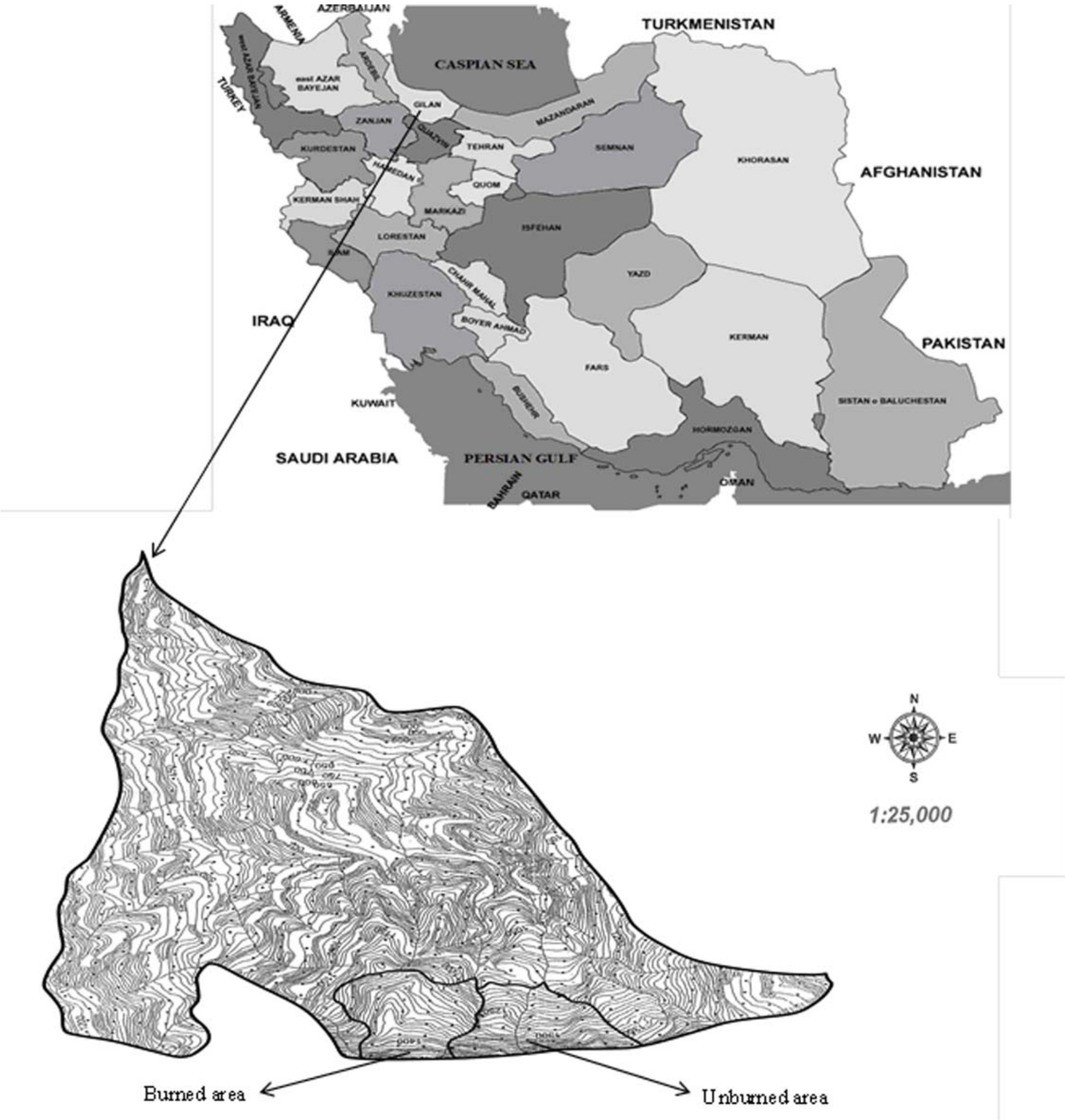


**Fig. 1 Study area map and location of plots**

**Table 1. Characteristics of the study area 37 years after the fire.**

| | Burned area | Unburned area | P |
|---|---|---|---|
| Slope, % | 30.5 | 33.1 | - |
| Elevation, m | 1295.6 | 1276.3 | - |
| Canopy cover, % | 81.1 | 86 | 0.028* |
| Herbaceous cover, % | 50.1 | 47.4 | 0.004* |
| Stand density, trees/ha | 1023 | 697 | 0.000* |
| Dead tree, tree/ha | 178 | 141 | 0.553 |
| Leaf litter depth | 7.05 | 8.82 | 0.014* |
| Mean DBH, cm | 20.2 | 34.61 | 0.000* |
| total basal area, $m^2$ | 44.43 | 87.46 | 0.000* |

* An asterisk indicates a significant difference between means at the $\alpha = 0.05$ level

The UB area had only *Fagus orientalis*, but the B area had four species including *Fagus orientalis, Carpinus betulus, Acer cappadocicum* and *Alnus glutinosa* (Table 2), although the dominant species was beech (*Fagus orientalis*), which accounted for 97.5% of the trees. Mean DBH for any individual species was largest (55.6 cm) in the B area and occurred in *Acer cappadocicum*. The smallest mean DBH for any species also occurred in the B area (19.7 cm) and occurred in *Fagus orientalis*, which had the greatest density (998 trees/ha) of any of the species in B or UB areas. Total number of trees in the B area was 1,023 compared to 697 trees/ha in the UB area. Overall Mean DBH in the B area was smaller (20.2 cm) than in the UB area (34.6 cm).

In the B area, there were twice as many trees in the 10 to 30 cm DBH classes as there were in the same classes in the UB area (Fig. 2). This relationship was reversed in the larger DBH classes, where trees were more numerous in the UB area. The predominance of trees in the B area (83.3%) occurred in the 10 to 30

cm DBH classes, where there was five times the number of trees compared to all other DBH classes. The largest tree DBH in the B area occurred in the 60 cm DBH class, whereas in the UB area, trees occurred in all DBH classes up to and including the 90cm DBH class.

**Table 2. Mean DBH (cm), total basal area, ($m^2$) and tree density (number/ha) of living trees in burned (B) and unburned (UB) areas 37 years after the fire.**

| Species | Family | Mean DBH (cm) | | Total basal area($m^2$) | | Density (trees/ha) | |
|---|---|---|---|---|---|---|---|
| | | B | UB | B | UB | B | UB |
| *Fagus orientalis* | Fagaceae | 19.7 | 34.6 | 40.7 | 87.4 | 998 | 697 |
| *Carpinus betulus* | Betulaceae | 31.2 | - | 1.7 | - | 15 | - |
| *Acer cappadocicum* | Aceraceae | 55.6 | - | 1.5 | - | 6 | - |
| *Alnus subcordata* | Betulaceae | 53.0 | - | | - | 4 | - |
| Total | | 20.2$^b$ | 34.6$^a$ | 44.4$^b$ | 87.4$^a$ | 1023$^a$ | 697$^b$ |

Different letters indicate significant differences between B and UB in each tree or stand metric (α = 0.05).

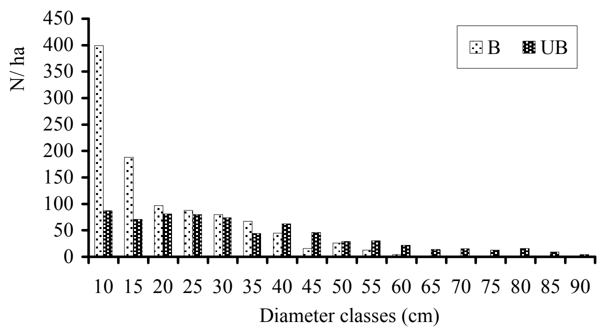


**Fig. 2 Diameter distribution for all species in burned (B) and unburned (UB) areas 37 years after the fire**

Shrub layer

There were four and five shrub species in the B and UB areas, respectively. Species such as *Ilex aquifolium, Ruscus hyrcanus* and *Vaccinium arctostaphylos* occurred in both areas while *Crataegus microphylla* was found only in the B area, and *Danae rasemosa* and *Rhamnus frangula* were only in the UB area (Table 3). Total number of shrubs for *Ilex aquifolium*, *Ruscus hyrcanus* and *Vaccinium arctostaphylos* were significantly greater ($p$≤0.05) in the UB area than in the B area. The density of shrubs in the UB area was four times greater than that in the B area.

Regeneration

Although *Alnus glutinosa* was present as mature trees only in the B area, there was a lack of regeneration in either area (Table 4). In contrast, regeneration of *Acer cappadocicum* occurred in both areas even though mature trees were only present in the B area. In both areas, *Fagus orientalis* dominated the regeneration such that it comprised 99% of all regeneration in the B area and 98% in the UB area. The density of regeneration was significantly less in the UB area ($p$≤0.05) in each size class. Total regeneration in B area was four times greater than in UB area. In the B area, half of the regeneration was in the DBH < 2.5 cm class, while in UB area 49% of the regeneration occurred in the < 1.3 m height class.

**Table 3. Density of shrub species (number/ha) in burned (B) and unburned (UB) areas 37 years after the fire.**

| Species | Family | Burned | Unburned | P |
|---|---|---|---|---|
| *Ilex aquifolium* | Aquifoliaceae | 192 | 731 | 0.010* |
| *Ruscus hyrcanus* | Asparaginaceae | 39 | 147 | 0.050* |
| *Vaccinium arctostaphylos* | Ericaceae | 17 | 103 | 0.050* |
| *Danae racemosa* | Liliaceae | - | 4 | - |
| *Rhamnus frangula* | Rhamnaceae | - | 1 | - |
| *Crataegus microphylla* | Asparaginaceae | 1 | - | - |
| Total | | 249 | 986 | 0.026* |

* An asterisk indicates a significant difference between means at the α = 0.05 level.

**Table 4. Density of regeneration (number/ha) by size class:1-Hieght < 1.3 m; 2-DBH < 2.5 cm; 3-DBH between 2.5 and 7.5cm in burned and unburned areas 37 years after the fire.**

| species | Height< 1.3 | | DBH<2.5 cm | | 2.5cm<DBH< 7.5cm | | Total | |
|---|---|---|---|---|---|---|---|---|
| | B | UB | B | UB | B | UB | B | UB |
| *Fagus orientalis* | 640 | 349 | 1480 | 280 | 782 | 95 | 2902 | 724 |
| *Carpinus betulus* | 21 | 14 | - | - | - | - | 21 | 14 |
| *Acer cappadocicum* | 9 | 2 | - | - | - | - | 9 | 2 |
| Total | 670$^a$ | 365$^b$ | 1480$^a$ | 280$^b$ | 782$^a$ | 95$^b$ | 2932$^a$ | 740$^b$ |

B--- Burned area; UB--- Unburned area

Different letters show significant difference between U and UB in each species.

## Discussion

Tree layer

In this study, fire decreased the percentage of canopy cover and increased the number of trees per hectare in the B area (Table 1). Increase in tree density occurred in the smaller DBH classes so that tree density below 30 cm in the B area was two times greater compared to the same size classes in the UB area. This result is just the opposite for mean DBH and basal area between the B and UB areas. The dominance of smaller diameter trees in the B area caused reductions in mean DBH and basal area compared to the UB area (Table 1). In a study conducted in Quebec forests, Smirnova et al. (2008) observed that the density of jack pine (*Pinus banksiana*) decreased but black spruce trees increased significantly after a fire. Similarly, the density of trees increased 228% 30 years after a fire in northern Arizona (Bataineh et al., 2006). In addition, they found that percentage of canopy cover and average diameter decreased in the burned area (Bataineh et al; 2006). Our results indicate that the forest has regenerated itself 37 years after fire and has become a younger forest with a

structure that is characterized by greater densities of smaller diameter (< 20 cm DBH) trees. However, the overall character of the forests structure is that of an uneven-aged forest with a reverse-J distribution, despite these minor shifts in the diameter distribution after the fire. *Fagus orientalis* has regenerated and maintained its dominance after the fire, which has also provided an opportunity for other species to add diversity to the forest.

The density of dead trees was not significantly different between the two areas (Table 1). Thus, any increase in dead trees due to the fire has become insignificant as snags have fallen and decayed over the past 37 years, and the growing space has been occupied by the ingrowth of saplings. Fire did not cause changes in the uneven-aged structure in the study area (Fig. 2), but it caused minor changes in species composition (Table 2). This change could be observed in the addition of three new species: *Carpinus betulus, Acer cappadocicum and Alnus glutinosa* that were not found in UB area. These species likely dispersed into the B area as seeds from adjacent forests, and they were able to develop in the increased light of canopy gaps where the fire had previously caused mortality of overstory trees. These species are shade intolerant and require higher levels of light than *Fagus orientalis* to grow and maintain dominance. However, *Fagus orientali*s was able to maintain dominance in the regeneration of the B area. In *Pinus jeffreyi* forests in northwestern Mexico (Stephens and Gill 2005) and in the Amazon forests of Brazil (Haugaasen et al 2003) it was found that the diameter distribution in burned areas resembled a reverse J or uneven-aged curve, suggesting that fire had not changed forest structure from preburn conditions.

Fire effects on species composition are highly variable. In our study, fire resulted in minor increases in tree species, with *Fagus orientalis* still dominating the forest 37 years after a fire. In contrast Wallenius et al. (2007) observed a significant historical change in species composition from *Pinus* spp. to *Picea abies* in southern Finland. In other research on the effects of fire on species composition in the forests of northern Iran, Banj-Shafiei et al. (2010) found that fire did change the species composition. This is in agreement with results obtained 46 years after fire in North West Uganda (Nangendo et al. 2010), where a change in tree species composition was observed.

Shrub layer

The fire significantly reduced ($p \leq 0.05$) the density of shrubs in the B area, such that their density was 1/4 of that in the UB area. In fact, fire had so negatively impacted the shrubs that they could not recover after the fire. Decreases in shrub density were most likely the result of competition with tree seedlings and saplings. *Ilex aquifolium*, *Ruscus hyrcanus* and *Vaccinium arctostaphylus* were the dominant shrub species both before and after the fire. In other words, the fire did not alter the domination pattern of these species. Although *Ilex aquifolium* suffered substantial reductions in density in the B area, it remained the most abundant species in the shrub layer in both areas. Brockway and Lewis (1997) found that fire significantly reduced *Ilex glabra* in the forests of Georgia, U.S. They further reported that *Danae racemosa* and *Rhamnus frangula* existed only in unburned areas and these species were unable to develop again after fire because fire likely destroyed the seed bank of these species. Roy and Sonie (1992) stated that the population of *Cistus salvifoliu*s and *monspeliensis* decreased 15 years after fire. Another study of fire from the coastal plains of South of Carolina, U.S. found that summer fires decreased shrub density or removed them even after 20 years (Lewis and Harshbarger, 1976). Elliott et al. (1999) showed that, although deciduous shrubs increased and evergreen shrubs decreased after fire, the total number of shrubs decreased to half of their preburn levels in the southern Appalachians, U.S.

Regeneration

After fire, regeneration of vegetation is based on the potential of individual species to reproduce in the new environment. There are two main mechanisms by which populations may establish in a burned site – vegetative regrowth of pre-existing individuals or establishment of new plants from seeds (Bond and Midgley 2001). The burned site is advantageous for Many species to establish new populations. The lack of competition from previously established plants and the increase of available resources (light from the canopy opening and post-fire flush of available nutrients) provide an advantage to species that have seed or plant material that survives the burn, or that are able to colonize the area from adjacent unburned regions (Bradstock 1992). Therefore, colonist species with long distance dispersal mechanisms and the ability to establish in open habitats are typically good candidates to initiate the regeneration process (Lloret and Zedler 2007).

The total density of regeneration was significantly greater in the B area (Table 4). Regeneration in the B area was four times more than in the UB area. Since *Fagus orientali*s, dominated the mature trees in the areas, they were able to produce prolific regeneration to maintain its prominence in both areas. Other factors that supported the dominance of *Fagus orientalis* regeneration and recruitment is the presence of existing saplings that could suppress new seedling reproduction, and the shade tolerance of the species that gives it a competitive advantage in small canopy gaps created by fire-caused mortality.

Most regeneration occurred in the < 2.5 cm DBH class suggesting that regeneration, which initiated after the fire, has been able to grow to sapling size over the past 37 years. Development of regeneration was promoted by greater available light resulting from a reduction in percentage of canopy cover (Table 1) in the B area. Rebertus and Burns (1997) and Hutchinson et al. (2005) reported that burning that caused open conditions in the forest allows regeneration to develop for a long time after burning.

Another way that fire can increase regeneration is by reducing the depth of litter. In the B area, the depth of litter was significantly decreased compared to the UB area (Table 1). Thus, seeds are better able to contact mineral soil, nutrients are released by the fire, more light is available through reduction of tree density, and decomposition of remaining organic matter provides more food germinating plants. Rinkes and McCarthy

(2007) found that more species establish in areas with limited litter than in areas with no or completely burned litters. In research conducted in the upland forests of the Cumberland Plateau, U.S., Royse et al. (2010) found that saplings increased after burning due to reduction in litter and more accessibility to light. These conditions permitted white oak (*Quercus alba*) to develop better in burned than in unburned areas.

Tree regeneration is favored by a decrease in shrub density, which provides more space and resources for saplings to grow. Martinez-Sanchez et al. (1998) and De Las Heras et al. (2002) reported that height growth and survival of *Pinus halepensis* saplings increased when shrubs were removed. In Sierra Nevada, U.S. mixed conifer forests, germination of *Pinus lambertiana* and *P. jeffreyi* in the burned area was greater than in the control area because of litter reduction, increase in mineral soil, and reduction of shrubs (Zald et al., 2008). Covington (2000) reported similar results. Thirty years after burning seedling establishment of *Pinus aristata* and *Pinus flexilis* increased, although there were differences between the two species depending on fire severity. Regeneration of these species continued over a long time. Fire promoted their regeneration by removing canopy cover shade and the litter layer (Coop and Schoettle, 2009). Research from monospecific to mixed forests after fire in Italy showed that seedlings of wind-dispersed trees such as maple species are found in the post-fire beech forest, however, their density is very low (van Gils et al., 2009). Green et al. (2010) studied regeneration on xeric sites on the Cumberland Plateau, U.S. following periodic burning, and found that burning improved the survival of *Quercus rubra* over its major competitor, *Acer rubrum*, although *Acer rubrum* had greater height growth than *Quercus rubra* or *Castanea sativa.*

## Conclusion

We assessed the long-term impacts of a wildfire on forests dominated by *Fagus orientalis* in north of Iran. We found that although fire did not change the overall uneven-aged structure of the beech forest it changed composition slightly by allowing several shade intolerant, pioneer species such as *Alnus glutinosa* to convert a pure *Fagus orientali*s forest into a more mixed forest with four species: *Fagus orientali*s, *Carpinus betulus, Acer cappadocicum* and *Alnus glutinosa.*

The considerable regeneration that has developed after the fire has helped to maintain the uneven-aged structure of the forest. The main reasons for increased tree regeneration were attributed to (1) reduction of litter depth, and (2) increase in available light from sun light and opening of the canopy and reduction in shrub competition. It is apparent that the forest is on a path to return to its natural state before the fire after 37 years.

Fire had positive impacts in our research, especially in promoting more diversity in tree regeneration, and thus it can be used to improve forest restoration. Fire suppression in the long-term causes reduction in diversity and regeneration. It is important to better understand fire impacts on forest structure and other components.

**Acknowledgements**

We thank Igor Drobyshev and Hein van Gils for helpful comments on an earlier version of the manuscript.

## Reference

Andariese SW, Covington WW. 1986. Change in understory production for three prescribed burns of different ages in Ponderosa Pine. *Forest Ecology and Management,* **14**: 193–203.

Anderson RC, Loucks OL, Swain AM. 1968. Herbaceous response to canopy over, light intensity, and through fall precipitation in coniferous forests. *Ecology*, **50**: 255–263.

Armour CD, Bunting SC, Neuenschwander LF. 1984. Fire intensity effects on the understory in Ponderosa Pine Forests. *Journal of Range Management*, **37**: 44–49.

Banj Shafiei A, Akbarinia M, Jalali G, Hosseini M. 2010. Forest fire effects in beech dominated mountain forest of Iran. *Forest Ecology and Management,* **259**: 2191–2196.

Bataineh AL, Oswald BP, Bataineh MM, Williams HM, Coble DW. 2006. Change in understory vegetation of a Ponderosa Pine forest in northern Arizona 30 years after a wildfire. *Forest Ecology and Management,* **235**: 283–294.

Bell DT, Hopkins AJM, Pate JS. 1984. Fire in the Kwongan. In: Pate, J.S., Beard, J.S. (Eds.), *Kwongan, plant life of the Sandplian*. Nedland: University of Western Australia Press, pp. 178–204.

Blackhall M, Raffaele E, Veblen TT. 2008. Cattle affect early-post fire regeneration in a *Nothafagus dombeyi-Austrocedrus Chilensis* mixed forest in northern Patagonia, Argentina. *Biological Conservation*, **141**: 2251–2261.

Bond WJ, Midgley JJ. 2001. Ecology of sprouting in woody plants: the persistence niche. *Trends in Ecology and Evolution,* **16**(1): 45–51.

Bradstock RA, Auld TD, Ellis ME, Cohn JS. 1992. Soil temperature during bushfires in semiarid. Mallee, shrublands. *Austral Journal of Ecology.* **17**, 433–440.

Brockway DG, Lewis, CE. 1997. Long-term effects of dormant season prescribed fire on plant community diversity, structure and productivity in a longleaf Pine wiregrass ecosystem. *Forest Ecology and Management,* **96**: 167–183.

Coates F, Lunt ID, Tremblay RL. 2006. Effects of disturbance on population dynamics of the threatened Orchid *Prasophyllum correctum* D.L. Jones and implications for grassland management in south eastern Australia. *Biological Conservation,* **129**: 59–69.

Connel JH. 1978. Diversity in tropical rainforest and topical reefs. *Science* **199**, 1302–1310.

Coop JD, Schoettle AW. 2009. Regeneration of Rocky Mountain bristlecone pine (*Pinus aristata*) and limber pine (*Pinus flexilis*) three decades after stand-replacing fires. *Forest Ecology and Management,* **257**: 893–903.

Covington WW. 2000. Helping western forests heel. Nature 408, 135–136.

De las Heras J, Martinez-Sanchez JJ, Gonzales-Ochoa AI, Ferrandis P, Herranz, JM. 2002. Establishment of *Pinus halepensis* Saplings following fire: Effects of competition with shrub species. *Acta Oecologica,* **23**: 91–97.

Elliott KJ, Hendrick RL, Major AE, Vose JM, Swank WT, 1999. Vegetation dynamics after a prescribed fire in the southern Appalachians. *Forest Ecology and Management,* **114**: 199–213.

Fernandez-Abascal I, Tarrega R, Luis-Calabuig E. 2004. Ten years of recov-

ery after experimental fire in a heathland: effects of sowing native species. *Forest Ecology and Management,* **203**: 147–156.

Fox MD, Fox BJ. 1988. The effect of fire frequency on the structure and floristic composition of a woodland understory. *Austral Ecology,* **11**: 77–85.

Gill AM. 1981. Adaptive response of Australian vascular plant species to fire. In: Gill, A.H., Groves, R.H., Noble, I.R. (Eds.), *Fire and the Australian Biota.* Canberra Australian Academy of Science, pp. 243–272.

Gonzales-Tagle MA, Schwendenmann L, Perez JJ, Schulz R. 2008. Forest structure and woody plant species composition along a fire chronosequence in mixed pine-oak forest in the Sierra Madre Oriental. *Forest Ecology and Management,* **256**: 161–167.

Green SR, Arthur MA, Blankenship BA. 2010. Oak and red maple seedling survival and growth following periodic prescribed fire on xeric ridge tops on the Cumberland Plateau. *Forest Ecology and Management,* **259**: 2256–2266.

Grime,JP. 2001. *Plant strategies vegetation processes and ecosystem properties, second ed*. Chichester: Wiley.

Hall GMJ, Wiser SK, Allen RB, Beets PN, Goulding CJ. 2001. Strategies to estimate national carbon stocks from inventory data: the 1990 New Zealand baseline*. Global Change Biology,* **7**: 389–403.

Harris GR, Covington WW. 1983. The effect of prescribed fire on nutrient concentration and standing crop of understory vegetation in ponderosa pine. *Canadian Journal of Forest Research,* **13**: 501–507.

Haugaasen T, Barlow J, Perez CA. 2003. Surface wildfires in central Amazonia: Short-term Impact on forest structure and carbon loss. *Forest Ecology and Management,* **179**: 321–331.

Herath DN, Lamont BB, Enright NJ, Miller BP. 2009. Impact of fire on plant-species persistence in post – mine restored and natural shrubland communities in southern Australia. *Biological Conservation,* **142**: 2175–2180.

Huang C, Marsh SE, Mclaran MP, Archer SR. 2007. Postfire stand structure in a semiarid savanna: cross-scale challenges estimating biomass. *Ecological Application,* **17**: 1899–1910.

Huston MA. 1994. *Biological Diversity: the coexistence of species on changing landscape*. Cambridge: Cambridge University Press. p.681.

Hutchinson TF, Sutherland EK, Yaussay DA. 2005. Effects of repeated prescribed fires on the structure, composition and regeneration of mixed-oak forest in Ohio. *Forest Ecology and Management,* **218**, 210–228.

Keeley JE, Lubin D, Fotheringham CJ. 2003. Fire and grazing impacts on plant diversity in the southern Sierra Nevada. *Ecological Application,* **13**: 1355–1374.

Krueger WC, Winward AH. 1974. Influence of cattle and big game grazing on understory structure of a douglas fir-ponderosa pine-kentucky bluegrass community. *Journal of Range Management,* **27**: 450–453.

Lamont BB, Marsula R, Enright NJ, Witkowfski ETF. 2001. Conservation requirements of an exploited wildflower: Modelling the effects of growing conditions, plant age and picking intensity. *Biological Conservation,* **99**: 157–168.

Laughlin DC, Bakker JD, Stoddard MT, Daniels ML, Springer JD, Gildar CN, Green AM, Covington WW. 2004. Toward reference conditions: Wildfire effects on flora in an old-growth Ponderosa Pine forest. *Forest Ecology and Management,* **199**: 137–152.

Levin SA, Paine RT. 1974. Disturbance, path information and community structure. In: Proceeding of Natural Academy of Science of the United States of America 71, 2744–2747.

Lewis CE, Harshbarger TJ. 1976. Shrub and herbaceous vegetation after 20 years of prescribed burning in the south Carolina Coastal Plain*. Journal of Range Management,* **29**: 13–18.

Lloret F, Zedler PH. 2009. The effect of forest fire on vegetation. In: (Eds.) Cerda, A., Robichaud, P.R. (eds.), *Fire effect on soils and restoration strategies*. Enfield, NH, USA: Science publisher, pp. 257－285.

Madany MH, West NE. 1983. Livestock grazing-fire regime interactions within montane forests of Zion National Park, Utah. *Ecology*, **64**: 661–667.

Martinez-Sanchez JJ, Ferrandis P, De las Heras J, Herrans JM. 1998. Influence of a *C.monspeliensis* on the early stages of development of *Pinus halepensis* seesdlings after fire in Tus Valley (SE Spain) In: Trabaud, L. (ed), *Fire Management and Landscape Ecology*. Washington: International Association of wildlandfire, pp. 205–211.

Mohadjer MR. 2006. *Silviculture*. Tehran: Tehran University Press, p. 387.

Moore MM, Casey CA, Bakker Jd, Springer JD, Fule PZ, Covington WW, Laughlin DC. 2006. Herbaceous vegetation response (1992-2004) to restoration treatment in a ponderosa pine forest. *Rangeland Ecology and Management,* **59**: 135–144.

Moreira F, Delgado A, Ferreira S, Borralho R, Oliveira N, Inacio M, Silva JS, Rego F. 2003. Effects of prescribed fire on vegetation structure and breeding birds in young *pinus pinaster* stands of Northern Portugal, *Forest Ecology and Management,* **184**: 225–237.

Moretti M, Barbalat M. 2004. The effects of wildfire on wood-eating beetles in deciduous forests on the southern slope of the Swiss Alps. *Forest Ecology and Management,* **187**: 85–103.

Nangendo G, Stein A, ter Steege H, Bongers F. 2005. Change in woody plant composition of three vegetation types exposed to a similar fire regime for over 46 years. *Forest Ecology and Management,* **217**: 351–364.

Rebertus AJ, Burns BR. 1997. The importance of gap processes in the development and maintenance of oak savannas and dry forests. *Journal of Ecol*ogy, **85**: 635–648.

Reinhardt ED, Ryan KC. 1988. Eight-year tree growth following prescribed under-burning in a Western Montata Douglas-fir/Western Larch Stand. United States of Department Agriculture. Forest Service. Intermountain Research Station.

Rinkes ZL, McCarthy BC. 2007. Ground layer heterogeneity and hardwood regeneration in mixed forest. *Applied Vegetati*on, **100**: 279–284.

Ross KA, Taylor JE, Fox MD, Fox BJ. 2004. Interaction of multiple disturbances: Importance of disturbance interval effects of fire on rehabilitating mined areas. *Austral Eco*logy, **29**: 508–529.

Roy J, Sonie L. 1992. Germination and population dynamic of *Cistus* species in relation to fire. *Journal of Application Ecology,* 29: 647–655.

Royse J, Arthur MA, Schorgendorfer A, Loftis DL. 2010. Establishment and growth of oak (*Quercus alba, Quercus prinus*) seedling in burned and fire-excluded upland forests on the Cumberland Plateau. *Forest Ecology and Management,* **260**: 502–510.

Sagheb-Talebi K, Sajedi T, Yazdian F. 2004. *Forests of Iran*. Tehran, Iran: Research Institute of forests and Rangelands, p. 28.

Smirnova E, Bergeron Y, Brais S. 2008. Influence of fire intensity on structure and composition of Jack Pine stands in the boreal forest of Quebec: Live trees, understory vegetation and dead wood dynamics. *Forest Ecology and Management,* **255**: 2916–2927.

Stephens S, Gill SJ. 2005. Forest structure and mortality in an old-growth Jeffrey Pine mixed conifer forest in north-western Mexico. *Forest Ecology and Management,* **205**: 15–28.

Taheri-Abkenar K, Pilehvar B. 2008. *Silviculture*. Iran: Hagh-Shenas Press. P. 296.

van Gils H, Odoi JO, Andrisano T. 2009. From monospecific to mixed forest

after fire? An early forecast for the montane belt of Majella, Italy. *Forest Ecology and Management,* **259**: 433–439.

van Mantgem P, Schwarts M. 2001. Monitoring fire effects for managed burns and wildfires: Coming to terms with Pseudorplication. *Natural Area*s *Journal*, **21**: 266–273.

Wallenius TH, Lilja S, Kuuluvaineh T. 2007. Fire history and tree species composition in managed *Picea abies* stands in southern Finland: Implication for restoration. *Forest Ecology and Management,* **250**: 89–95.

Wardle JA. 1984. The New Zealand beeches-ecology, utilization and management. New Zealand Forest Service, Wellington, N.Z.

Whelan J. 1995. *The ecology of fire*. Cambridge: Cambridge University Press, p. 346.

Wright HA, Bailey AW. 1982. *Fire Ecology: United States and southern Canada*. New York, NY: John Wiley and Sons, p. 501.

Yang J, He HS, Shifley SR. 2008. Spatial controls of occurrence and spread of wildfires in the Missouri Ozark highlands. *Ecological Application,* **18**: 1212–1225.

Zald HSJ, Gray AN, North M, Kern RA. 2008. Initial tree regeneration responses to fire and thinning treatments in a Sierra Nevada mixed conifer forest. USA. *Forest Ecology and Management,* **256**: 168–179.

Zimmerman, GT, Neuenschwander LF. 1984. Livestock grazing influences on community structure, fire intensity, and fire frequency within the Douglas-fir/ninebark habitat type. *Journal of Range Management*, **37**: 104–110.